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BIOACCUMULATION AND EFFECTS OF CHLORINATED DIBENZODIOXINS AND FURANS IN FISH, SHELLFISH AND CRUSTACEA. A BRIEF REVIEW

Derek C.G. Muir
Research scientist,
Dept. of Fisheries and Oceans, Winnipeg.

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Introduction:

There is a growing body of literature on bioaccumulation and toxicity of chlorinated dibenzodioxins (PCDDs) and furans (PCDFs) to aquatic organisms. Most of the toxicity studies have been conducted on freshwater fish (rainbow trout, carp, perch). There appears to be no information on the toxicity of PCDD/PCDF to crustacea or shellfish although there is some experimental bioaccumulation data for these organisms.

No review of toxicity of PCDD/PCDF is complete without consideration of the bioavailability of these compounds and the pathways by which they are accumulated in aquatic organisms. This review therefore considers pathways of accumulation and acute and chronic toxicity to aquatic organisms.

(1) Pathways of accumulation: PCDD/PCDFs are extremely hydrophobic compounds (water solubility <500 ng/L for those with 4 or more chlorine substituents) and are associated with suspended and particulate matter in natural waters. For example, in $40m^3$ lake enclosures spiked with 1,3,6,8-T_4CDD (which has similar water solubility to 2,3,7,8-T_4CDD), <15% was present in solution; >85% was adsorbed to particulate or dissolved organic matter (Servos 1988). The fact that PCDD/PCDF are not detected in open lake waters in the Great Lakes (detection limits 10^{-3} ng/L), although they are detected in fish and sediments at ng/kg levels, further illustrates the degree of hydrophobicity.

Since PCDDs/PCDFs will enter into and be transported in aquatic environments bound to dissolved or suspended material i.e from pulp mill effluents, dump sites or the atmosphere, sediment dwelling and detrital feeding aquatic organisms will be directly exposed. Perhaps the best studies to date on bioavailability of PCDD/PCDFs from sediment have been reported by Kuehl et al. (1987) who showed that 2,3,7,8-substituted PCDD and PCDF congeners in Wisconsin river sediments were accumulated by carp during a 55 day exposure period. 2,3,7,8-T_4CDD was present in sediments at 39 pg/g but was undetectable in centrifuged water samples. Kuehl et al (1987) concluded that uptake of 2,3,7,8-T_4CDD could be explained entirely by ingestion of contaminated particulates by carp.

Lake trout and other predatious freshwater and marine fish are believed to accumulate PCDDs and PCDFs entirely from food chain transfer. Only the 2,3,7,8-substituted tetra- and pentachloro dioxins, and tetra- to hexachloro furans, appear to biomagnify, i.e. increase in concentration up the food chain, based on monitoring data. Octachlorodioxin concentrations actually decline up the food chain from detritus to lake trout (Muir et al. 1988). The extent of food chain transfer and biomagnification can be assessed from knowledge of the assimilation efficiency of the pollutant from food and elimination rate or half-life in the fish (Thomann and Connolly 1985). Assimilation efficiencies of 2,3,7,8-substituted PCDD/PCDFs from food are relatively high (30 - 50%) (Kleeman et al. 1986aa and b; Muir et al. 1988) except for octachlorodioxin and furan (Niimi and Oliver 1986; Muir et al. 1986). Elimination rates for 2,3,7,8-T_4CDD are slow with half-lives of about 100 days in rainbow trout following uptake from food (Kleeman et al. 1986a). These values for assimilation and elimination rate are similar to those for PCBs and indicate that uptake from food (via the food chain) will account for >99% of the body burden of 2,3,7,8-substituted PCDD/PCDFs in lake trout in the Great Lakes. These predictions (which are based on the food chain transfer model of Thomann and Connolly 1985) are confirmed by monitoring

data for 2,3,7,8-T_4CDD and T_4CDF in the Great Lakes (Stalling et al. 1983) and in the Baltic Sea (Rappe et al. 1987).

There is a wide variation in the reported half-lives for 2,3,7,8-T_4CDD in fish. For example, Mehrle et al. (1988) reported half-lives of 15 days for 2,3,7,8-T_4CDD in fingerling rainbow trout following aqueous exposure while Kleeman et al (1986a) reported half-lives of 105 days for the same species after uptake from food. Differences between in half-lives between the food and gill uptake routes may reflect different routes of transport and storage of 2,3,7,8-T_4CDD following diffusion across membranes of the gill or the GI tract. Half-lives obtained following food ingestion or oral exposure are probably the most realistic because accumulation from food is the most important pathway for aquatic organisms

(2) Acute and chronic toxicity: Lethal doses of 2,3,7,8-T_4CDD to fish appear to be in the same range as those for sensitive mammalian species (Table 1). Assuming complete assimilation of the dose the lethal levels are (on a whole body basis) at least 100 times higher than observed in monitoring of 2,3,7,8-T_4CDD in whole fish from contaminated sites in the Great Lakes (Stalling et al. 1983; Baumann and Whittle, 1988).

PCDDs and PCDFs are found mainly in the fatty tissues of fish. Kleeman et al.(1986a and b) found highest concentrations of 2,3,7,8-T_4CDD in visceral fat and pyloric caeca of rainbow trout and perch following 91 days feeding. Lowest concentrations were in the skeletal muscle (skinless fillet). Kleeman et al. (1986a and b) reported no effects on growth of rainbow trout or perch a diet of 494 ng of 2,3,7,8-T_4CDD /kg food although visceral fat contained 3270 ng/kg of the compound (whole body residues were 250 ng/kg). Whole fish concentrations of PCDD/PCDFs will tend to be higher, by a factor roughly proportional to lipid content of the tissue, than in the fillet (Branson et al. 1985; Kuehl et al. 1987).

Reports of PCDD/PCDF concentrations in fatty tissues are generally not a good indicator of potential effects on the organism unless they can be converted to a whole body residue. An interesting case is the crab hepatopancreas which is also a high lipid tissue. If we make the assumption that the LD_{50}'s for this species are in the same range as those in Table 2, then ug/kg concentrations of 2,3,7,8-T_4CDF in hepatopancreas would not be lethal but simply reflect the storage of PCDF in fatty tissues. Whole body residues would be expected to be 1/10 to 1/100th of the levels in heptatopancreas or well under the expected lethal dose. Nevetheless ug/kg concentrations in fatty tissues indicate extraordinarily high concentrations of 2,3,7,8-T_4CDF in the diet of the crabs.

A few of the sublethal effects of 2,3,7,8-T_4CDD on fish following dietary, water or intraperitoneal (ip) exposure are listed in Table 2. 2,3,7,8-T_4CDD has been shown to be highly toxic to freshwater fish when administered via water at ug/L and ng/L concentrations (Miller et al.1979; Helder 1980; 1981). This early work was limited by lack of measurements of TCDD concentrations in the organisms or test water. Table 2 also includes where possible, the concentrations in the test species so that effects can be related to whole fish concentrations. This should enable qualitative assessment of the effects on fish with the residue levels observed in monitoring studies.

The most readily observed sublethal effect is reduced rate of growth of the exposed population. Individual fish experience weight loss, fin

darkening and erosion, hemorrhaging particularly along the lateral line, exopthalmia, and behavioral changes (Mehrle et al. 1988; Cook et al. 1986; Johnson et al. 1986). Effects are observed where individuals have > 1000 ng/kg of 2,3,7,8-T_4CDD on a whole fish (wet wt) basis. For example, rainbow trout exposed to 0.038 ng/L of 2,3,7,8-T_4CDD for 28 days had 980 ng/kg concentrations and showed reduced growth during the depuration phase of the experiment but no increase in mortality. Rainbow trout fed 494 ng/kg for 91 days showed no effects (growth, fin erosion, and cutaneous hemorrhage were investigated). After 91 days the juvenile trout contained 250 ng/kg (whole fish) of 2,3,7,8-T_4CDD (total tritium labelled compound)(Kleeman et al. 1988). The studies of Kleeman et al (1986a and b) establish a lower limit for readily observed sublethal effects on fish however they do not address effects such as carcinogenicity, teratogenicity and reproductive toxicity which are known to occur in mammals exposed to 2,3,7,8-T_4CDD and T_4CDF at levels lower than those causing weight loss.

Fish embryos are extremely sensitive to 2,3,7,8-T_4CDD. Cooper et al. (1986) used the embryo of the Japanese medaka (Oryzias latipes) as a bioassay for dioxin contaminated soils. Helder (1981) showed that exposure of rainbow trout and pike eggs to 2,3,7,8-T_4CDD at 0.1 ng/L concentrations caused effects on the developing embryos (hemorrhaging and edema). These concentrations are unrealistically high, however, and no recent work at sub-picogram/L concentrations has been reported.

At the biochemical level, induction of mixed function oxidase (MFO) enzymes in rainbow trout liver by 2,3,7,8-T_4CDD has been reported (Vodicnik et al. 1981)(Table 2). MFO enzyme induction by PCDDs and PCDFs in rat liver has been studied extensively and correlations with reduced weight gain and thymic atrophy (in rats) have been reported (Mason et al. 1985). No similar correlations exist at present for fish. MFO enzyme induction in wild fish has been correlated with PCB concentrations (Luxon et al. 1987; Stegeman et al. 1987) but not with PCDD/PCDF levels. Lockhart (1988) reported elevated levels of MFO activity (ethoxyresorufin O-deethylase) in chinook salmon smolts collected downstream from a bleached kraft mills near Quesnel and Prince George, B.C. There is great potential for this technique as an indicator for PCDD/PCDF exposure, for example in caged fish in pulp mill effluents.

There is less information on the toxicity of PCDFs to fish. The work of Mehrle et al. (1988) indicates that 2,3,7,8-T_4CDF is less toxic to rainbow trout than 2,3,7,8-T_4CDD by a factor of about 10 in the case of reduced growth. This result is consistent with the mammalian toxicology literature of 2,3,7,8-T_4CDF which indicates that T_4CDF is about 10-fold less potent than T_4CDD in terms of enzyme induction, teratogenicity and weight loss. Other PCDF congeners, especially 2,3,4,7,8-P_5CDF are of equal or greater toxicity to mammalian species than the 2,3,7,8-T_4CDF congener (Mason et al. 1985; Brewster et al. 1988). As yet there is no published information of the toxicity of other PCDFs to aquatic organisms. Preliminary studies in our labs (DFO Winnipeg) on the oral exposure of rainbow trout 2,3,4,7,8-P_5CDF indicated no lethality over a 21 day period (Table 1) however elevated MFO (EROD) levels were observed following a single oral dose of 5000 ng/kg of the compound (Muir et al. 1988b).

Summary

The information available on the toxicity and effects of PCDDs and PCDFs to aquatic organisms is limited mainly to freshwater fish. There is, however, a sufficient data base at least for 2,3,7,8-T_4CDD to make some educated guesses as to what body burden will result in sublethal effects (such as reduced growth) in aquatic organisms in general. No effect levels of exposure for 2,3,7,8-T_4CDD via food, which is widely viewed as the the major pathway of exposure of predatious and detritivorous fish, are <494 ng T_4CDD per kg food (494 parts/trillion). Concentrations of 2,3,7,8-T_4CDD in whole fish which are greater than 1000 ng/kg are associated with sublethal effects such as reduced growth. Effects of 2,3,7,8-T_4CDF occur at about 10-fold higher concentrations. These effects levels do not take into account the possible additive effect of the variety of toxic 2,3,7,8-substituted tetra to hexachloro dioxins and furans which are often seen in fish from contaminated sites. There is clearly a need for more information on sublethal effects such as reproductive or hatching success and for correlations of MFO enzyme induction in fish liver with sublethal effects. These latter effects may occur at whole fish concentrations of 2,3,7,8-T_4CDD substantially below 1000 ng/kg.

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Table 1. Comparison of LD50's for 2,3,7,8-T_4CDD in fish and mammals[1].

Species	LD_{50} (ug/kg)	Route of Exposure[2]	Time period (days)	Reference
Rainbow trout	5 - 25	ip	80	a
Perch	1 - 5	ip	80	a
Rat (male)	22	oral	30	b
Mice	284	oral	30	c
Rabbits	115	oral	30	b
Guinea pig (male)	0.6	oral	30	c
Rhesus monkey	<70	oral	35	d

[1] References. (a) Kleeman et al. 1986c. (b) Esposito et al. 1980 (c) McConnell et al. 1978a (d) McConnell et al. 1978b

2 Single doses.

Table 2. Summary of recent toxicity studies of 2,3,7,8-substituted PCDD/PCDF on aquatic organisms.

Species	Compound	Route of uptake	Dose ng/kg or ng/L	Exposure time days	Concentration in whole fish ng/kg wet wt	Effect or LD50	Reference
PCDDs							
Rainbow trout	2378-T_4CDD	food	494	91	250	no effect	Kleeman et al (1986a)
Perch	2378-T_4CDD	food	494	91	143	no effect	Kleeman et al (1986b)
Carp	2378-T_4CDD	sediment	39	55	7.5	no effect	Kuehl et al. (1987)
Rainbow trout	123478-H_6CDD	food	100000	30	7000	Possible growth reduction	Muir et al. (1988)
Rainbow trout	1234678-H_7CDD	food	100000	30	2000	No effect	
Rainbow trout	2378-T_4CDD	water	0.038	28	980	reduced growth	Mehrle et al (1988)
			0.176	28	4520	growth,behavior,	
			0.382	28	10950	and mortality at	
			0.789	28	>15400[1]	at 3 higher conc.	
Carp	2378-T_4CDD	water	0.100	71	~2000	Wt loss other chronic effects	Cook et al. (1986)
Rainbow trout	123478-H_6CDD	water	10	5	3500	No effect	Muir et al. (1985)
Rainbow trout	1234678-H_7CDD	water	11	5	2500	No effect	
Fathead minnows	2378-T_4CDD	water	0.8-1.4	28	25000	delayed mortality	Adams et al. (1986)
Daphnia magna	2378-T_4CDD	water	1-1000	2	ND[2]	no effect	
Rainbow trout	2378-T_4CDD	ip	1200	single dose	ND	elevated MFO activity	Vodicnik et al.(1981)
Rainbow trout	2378-T_4CDD	ip	1000-125000	single dose	ND	80 day LD50 = 5000-25000 ng/kg	Kleeman et al (1986c)
Perch	2378-T_4CDD	ip	1000-125000	single dose	ND	80 day LD50 = 1000-5000 ng/kg	

Table 2 continued

PCDFs						
Atlantic salmon	T_4CDF[3]	food	2.2×10^6	140	<20000	50% mortality in 120 days — Zitko and Choi (1973)
Rainbow trout	2378-T_4CDF	water	0.41 3.93	28 28	2480 9650	no effect on growth mortality, growth effects — Mehrle et al. (1988)
Rainbow trout	23478-P_5CDF	food	5000	single dose	2000	MFO induction, no growth effect at 21 d — Muir et al. (1988b)

[1] Concentration was 15400 ng/kg at 21 days.
[2] Not determined.
[3] Unspecified di, tri, tetra and octachlorofuran mixture.